Pairing high-frequency data with a link-node model to manage dissolved oxygen impairment in a dredged estuary

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Abstract

High-frequency data and a link-node model were used to investigate the relative importance of mass loads of oxygen-demanding substances and channel geometry on recurrent low dissolved oxygen (DO) in the San Joaquin River Estuary in California. The model was calibrated using 6 years of data. The calibrated model was then used to determine the significance of the following factors on low DO: excavation of the river to allow navigation of large vessels, non-point source pollution from the agricultural watershed, effluent from a wastewater treatment plant, and non-point source pollution from an urban area. An alternative metric for low DO, excess net oxygen demand (ENOD), was applied to better characterize DO impairment. Model results indicate that the dredged ship channel had the most significant effect on DO (62 % fewer predicted hourly DO violations), followed by mass load inputs from the watershed (52 % fewer predicted hourly DO violations). Model results suggest that elimination of any one factor will not completely resolve DO impairment and that continued use of supplemental aeration is warranted. Calculation of ENOD proved more informative than the sole use of DO. Application of the simple model allowed for interpretation of the extensive data collected. The current monitoring program could be enhanced by additional monitoring stations that would provide better volumetric estimates of low DO.

Keywords: San Joaquin River, Estuary, Dissolved oxygen, Water quality, TMDL, California, Models

Introduction

Like other impacted estuaries in the world, chronic and recurrent low dissolved oxygen (DO) conditions have persisted in the tidal portion of the San Joaquin River (SJR) for decades (McCarty 1969; US ACE 1988). Inadequate DO has led to the loss of fisheries and impairment of the river as a water supply for domestic, agricultural, and industrial purposes (Hallock et al. 1970; Stringfellow et al. 2009). In response, regulators have placed sections of the SJR on the 303(d) list for non-compliance with DO standards and calculated the total maximum daily load (TMDL) (Gowdy and Grober 2005). Resolution of low DO conditions in the SJR has become a major focus of ecosystem restoration efforts and TMDL implementation in California (Foe et al. 2002; Gowdy and Grober 2005; Stringfellow 2008b; Stringfellow and Camarillo 2014; Lee and Jones-Lee, 2003).

In some respects, development of anoxic conditions in the SJR Estuary is similar to what is observed in other regions (Cloern 2001; Shenk and Linker 2013; Bianchi et al. 2010). Watershed-derived anthropogenic nutrients contribute to eutrophication and result in decreased DO (Howarth et al. 2002). Unique aspects of the SJR, however, make delineation of responsibility—an essential part of the TMDL process—challenging. Unlike many other estuaries, the section of the SJR most affected by low DO—the Deep Water Ship Channel (DWSC)—is a freshwater estuary that has been dredged and channelized for navigation (US ACE 1988). Although water quality problems persist throughout the SJR, low DO is most prominent in the estuary (Gowdy and Grober 2005). The sources of this impairment are well documented although the relative contributions of these sources are not clearly understood (Foe et al. 2002).

Contributing to DO impairment are large quantities of anthropogenic nutrients and organic materials that lead to eutrophication (Ohte et al. 2007; Stringfellow et al. 2009; Gulati et al. 2016). Within the watershed, anthropogenic nutrients are primarily derived from agriculture (Stringfellow 2008a; Gowdy and Grober 2005). These nutrients cause excessive phytoplankton growth during the hot and dry summers, which contribute to mass loads of oxygen-demanding substances (ODS) observed along the main stem of the river (Ohte et al. 2007; Volkmar and Dahlgren 2006; Jassby and Van Nieuwenhuyse 2005). The semi-arid climate of Central California results in abundant sunlight that fuels phytoplankton growth as well as high water temperatures that limit DO capacity (Jassby and Van Nieuwenhuyse 2005; Lee and Jones-Lee, 2003). Phytoplankton growth is further stimulated by the river's shallow depth (Leland 2003). Phytoplankton (measured by chlorophyll *a*) correlates strongly with biochemical oxygen demand (BOD) (Volkmar and Dahlgren 2006; Ohte et al. 2007). Terrestrial carbon also appears to contribute to watershed-derived ODS (HydroQual 2006b).

Other sources of DO impairment are the City of Stockton Regional Wastewater Control Facility (RWCF) and urban runoff from the City (Lehman et al. 2004; Gowdy and Grober 2005; Foe et al. 2002; Lee and Jones-Lee 2003). The combination of nutrient and oxygen-demanding inputs from agricultural and urban sources is typical in TMDLs developed in other watersheds (Howarth et al. 2002; US EPA 2008). Ammonia from point and non-point sources has been identified as a major contributor to low DO (Lehman et al. 2004; Jassby and Van Nieuwenhuyse 2005). Historically, effluent from the non-nitrifying RWCF contained high ammonia concentrations (McCarty 1969; US ACE 1988) although upgrades to the facility have decreased the amount of ammonia discharged. The extent to which urban runoff contributes to ODS in the DWSC is unknown (Lee and Jones-Lee, 2003; US ACE 1988). In November 2002, high ODS concentrations were noted in stormwater discharged from the City during a large storm, and subsequently, later that winter the DWSC became hypoxic (DO ≤2 mg L^{-1}) and fish kills were observed (Lee and Jones-Lee, 2003).

Dredging of the SJR Estuary has also contributed to low DO. The deepened channel limits the proportion of the water column in the photic zone, which limits photosynthesis and the resulting DO production (Jassby and Van Nieuwenhuyse 2005). Due to light limitations, phytoplankton from the SJR undergoes massive respiration within the DWSC, which consumes DO (Stringfellow et al. 2009; Leland 2003; Jassby 2005). The deepened channel has resulted in long residence times of 4 to 30 days within the DWSC (Lee and Jones-Lee, 2003; Schmieder et al. 2008), further exacerbating the respiration of phytoplankton. Residence times are longer in summer and fall when river flows are low due to increased agricultural diversions and decreased regulated flows from reservoir releases. The result is that low DO is related to river flow (Jassby and Van Nieuwenhuyse 2005; Gowdy and Grober 2005), although the relationship is complex and requires further study (Jones & Stokes 2006). The deepened ship channel also limits reaeration due to the low surface area to volume ratio (Foe et al. 2002).

To increase DO in the DWSC and meet TMDL objectives, extensive monitoring and modeling has been done to link the sources of pollution with water quality outcomes (Stringfellow et al. 2009; Monismith et al. 2008; Lehman et al. 2004; Ohte et al. 2007; Volkmar and Dahlgren 2006; Stringfellow and Camarillo 2014; Gulati et al. 2016). Since the biological, chemical, and physical processes within the SJR are interactive and complex, modeling has been used to delineate the sources of low DO. Preliminary models have included a statistical model (Jassby and Van Nieuwenhuyse 2005), a Streeter-Phelps type model (Foe et al. 2002), and a box model (Lee and Jones-Lee, 2003). One-dimensional (1-D) models used to study DO in the SJR include a link-node model developed by Resource Management Associates (RMA), a different link-node model developed by Systech, and the California Department of Water Resources (DWR) Delta Simulation Model II with a water quality module (DSM2-QUAL) (Jones & Stokes 2006; Schanz and Chen 1993; Chen and Tsai 1997, 2002; Jones & Stokes 2005a; CDWR 2013c; King 1996; RMA 1988; US ACE 1988). Three-dimensional (3-D) models used include the ECOMSED model by HydroQual (2006a, b) and the SI3D hydrodynamics model with integrated water quality capability (SI3DWQ), as developed by researchers from UC Davis, Stanford, and the US Geological Survey (USGS) (S. Monismith et al. 2008; Schladow and Monismith 2009). Application of the SI3D model has been limited, however, and the model has not yet been extended to simulate DO (Monismith et al. 2008; Schladow and Monismith 2009; Doyle 2010). In response, developers of the SI3D model used a 1-D model for temperature (Monismith et al. 2009).

In 2005, a panel of experts convened and concluded that 1-D models are currently better suited for study of low DO in the SJR and development of TMDL decisions (Jones & Stokes 2006). Although 3-D models have research value, the panel felt that they were not inherently more accurate or useful for predicting low DO in the DWSC (Jones & Stokes 2006). The suitability of 1-D models in the SJR is likely related to the fact that although the DWSC may stratify, vertical stratification is not stable and the DWSC mixes on a daily basis, even in summer (Lehman et al. 2004; Monismith et al. 2008; Schmieder et al. 2008; Spier et al. 2013). Both Monismith (2008) and HydroQual (2006a) noted weak vertical stratification in the DWSC, on the order of 1–3 °C, which further suggests diurnal mixing. The instability of vertical stratification likely makes 3-D modeling challenging since it can be difficult to predict formation and dissolution of stratified layers. Based on its history of use in a regulatory context (Chen & Tsai 1997, 2002; Lee & Jones-Lee, 2003), its scientific basis, previous evaluation in two peer review processes including one completed by the US EPA (Chen & Tsai 2002), and incorporation of a graphical user interface, the peer review panel determined that the Systech Link-Node model was best suited for DO TMDL management in the DWSC (Jones & Stokes 2006). The work presented here represents the subsequent application of this model.

Compared with previous studies, the goal of this project was to model DO over a long time period (6 years) using a small time step (1 h). Recent studies have shown the value of high-frequency data for studying water quality processes that fluctuate diurnally and for identifying sources of impairment (Halliday et al. 2015; Lanoux et al. 2013). Here, the calibration data set used covered a longer period of time than what had been used previously (HydroQual 2006a; Schladow and Monismith 2009; Schanz and Chen 1993; Chen and Tsai 2002) and the time step was smaller than in previous studies (Jassby and Van Nieuwenhuyse 2005). High-frequency data were available (15-min increments) as was extensive grab sample data for the upstream SJR. The contribution of this work was to quantify the relative importance of previously identified sources of DO impairment. In addition, regulators introduced the metric of excess net oxygen demand (ENOD) for evaluating low DO and allocating responsibility (Gowdy and Grober 2005). A goal here was to evaluate this metric. ENOD represents the mass quantity of DO deficit calculated when DO is below the regulatory standard such that the river's assimilative capacity has been exceeded.

A 1-D model was calibrated for the SJR Estuary, establishing a baseline. The calibrated model was then used to quantify the impact of the following: (1) changes in channel depth due to the DWSC, (2) contributions of ODS from non-point sources in the upstream reaches of the SJR, (3) contributions of ODS from the Stockton RWCF, and (4) contributions of ODS from urban tributaries. In each scenario, one of the four factors contributing to low DO was removed so that the simulation results could be compared with the baseline. Since flow is important, the flow rates were not altered and were maintained while ODS mass loads were removed. Conducting the analysis over a 6-year period allowed for inclusion of various flow rates. The results were used to determine if ENOD is a valuable metric of low DO.

Materials and methods

Site description

The SJR originates in the Sierra Nevada Mountains, descends west to the San Joaquin Valley floor, and then flows north (Fig. 1). The SJR joins the Sacramento River to form the Sacramento-San Joaquin River Delta (Delta), a coastal estuary that discharges into San Francisco Bay (Bay). Water quality issues in the SJR have repercussions for the entire Bay-Delta ecosystem (Kimmerer 2004). The Delta provides drinking water for 25 million people (CVRWQCB 2011) and delivers irrigation water for California's US$45 billion agricultural industry (USDA 2013). The SJR basin covers an area of 41,000 km^2; approximately two thirds of the basin is mountainous and forested while the remainder consists of a flat valley floor where irrigated agriculture is the predominant land use (CDWR 2013c). The population in the region is approximately 2 million (CDWR 2013d). On the west side, soils are of marine origin with high mineral content, while east-side soils are derived of weathered granite (Gronberg et al. 1998). Differences in geology cause differences in water quality of the tributaries.

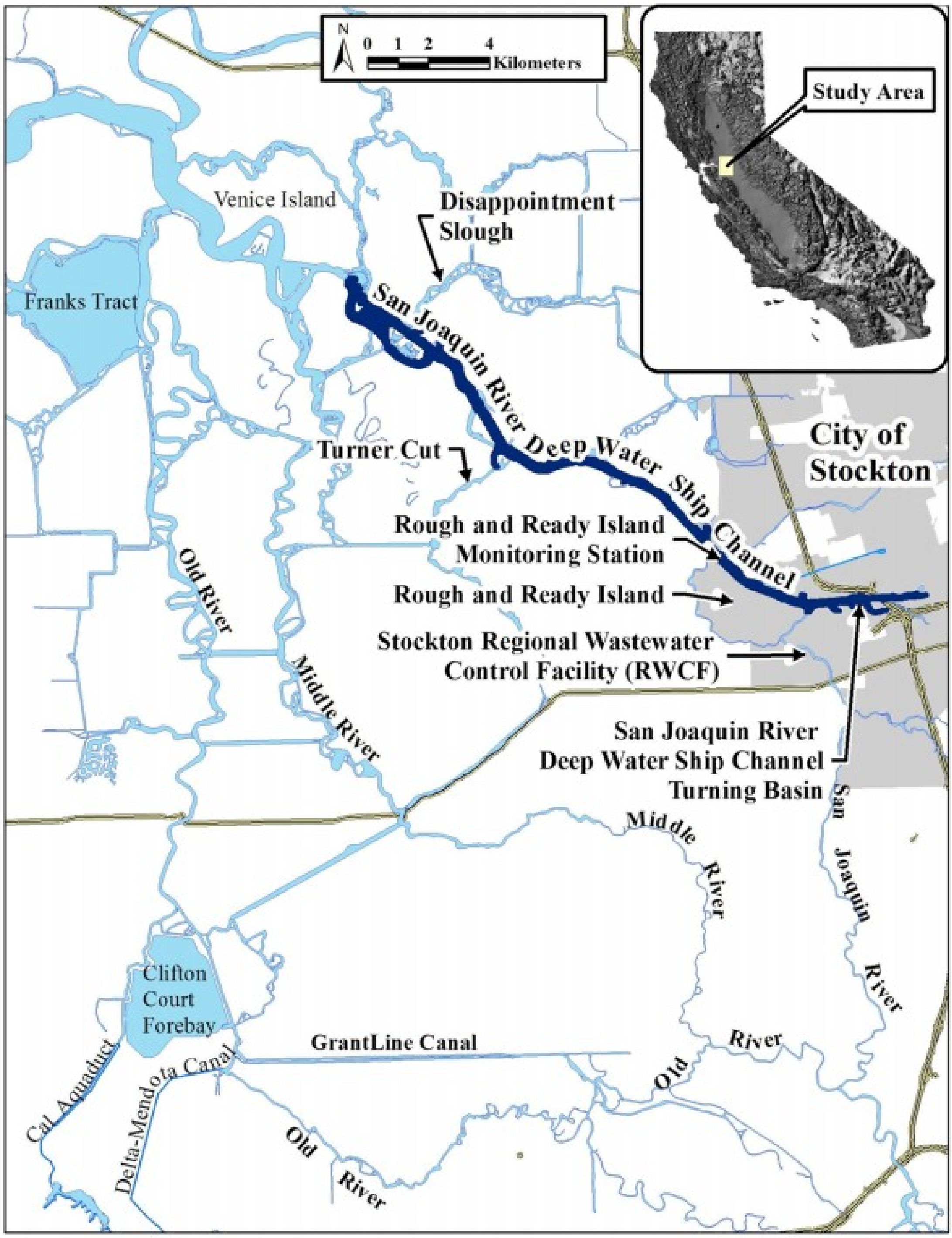

Fig. 1 San Joaquin River in Central California

The hydrologic cycle of the SJR basin is predominately artificial (Galloway and Riley 1999). Eighty-five percent of historic wetlands have been drained and converted to agriculture (SJVDP 1990). Flow in the SJR and its tributaries is highly regulated (Stringfellow and Jain 2010). While 90 % of precipitation in the valley occurs from November to April, water demand primarily occurs during the growing season (June through September). Reservoirs on the main tributaries (Stanislaus, Tuolumne, Merced) and upstream river collect and store rainwater and snow melt from the Sierra Nevada Mountains during the winter and spring months. Reservoir releases during the growing season provide water for agriculture and other uses. Diversions are located along the length of the river—including those to state and federal water projects—as are river inputs that include agricultural drainage (Gowdy and Grober 2005). Diversions can cause net negative flow in the river.

Downstream of Vernalis, the SJR is tidal. The SJR runs adjacent to the City of Stockton, and a section of the riverine estuary—the DWSC—has been dredged to permit navigation of ocean-going vessels from San Francisco Bay into the Port of Stockton. The DWSC is approximately 11 km long, 10.7 m deep, and 150–300 m wide; in contrast, the SJR upstream of the channel is approximately 1.5–3.0 m deep and 46–76 m wide (Gowdy and Grober 2005; US ACE 1988). Effluent from the City of Stockton RWCF discharges into the SJR just upstream of the DWSC, while urban runoff is conveyed into the DWSC via urban tributaries (Fig. 1). The riparian areas along the DWSC are highly altered to accommodate urban development, flood control, and navigation.

Model description

The 1-D SJR-Link-Node model was used to simulate flow and water quality in the Estuary (Chen and Tsai 2002). The model is described briefly here; more detail is provided in Supplemental Materials Sections S1 and S2. The upstream model boundary is just below the confluence with Old River, and the downstream boundary is at Disappointment Slough (Fig. 2). An Euler grid system is used, and the river is divided into segments (nodes) that have bi-directional connections (links), simulating tidally influenced flow and mass transport. The model domain contains 109 nodes (Chen and Tsai 2002).

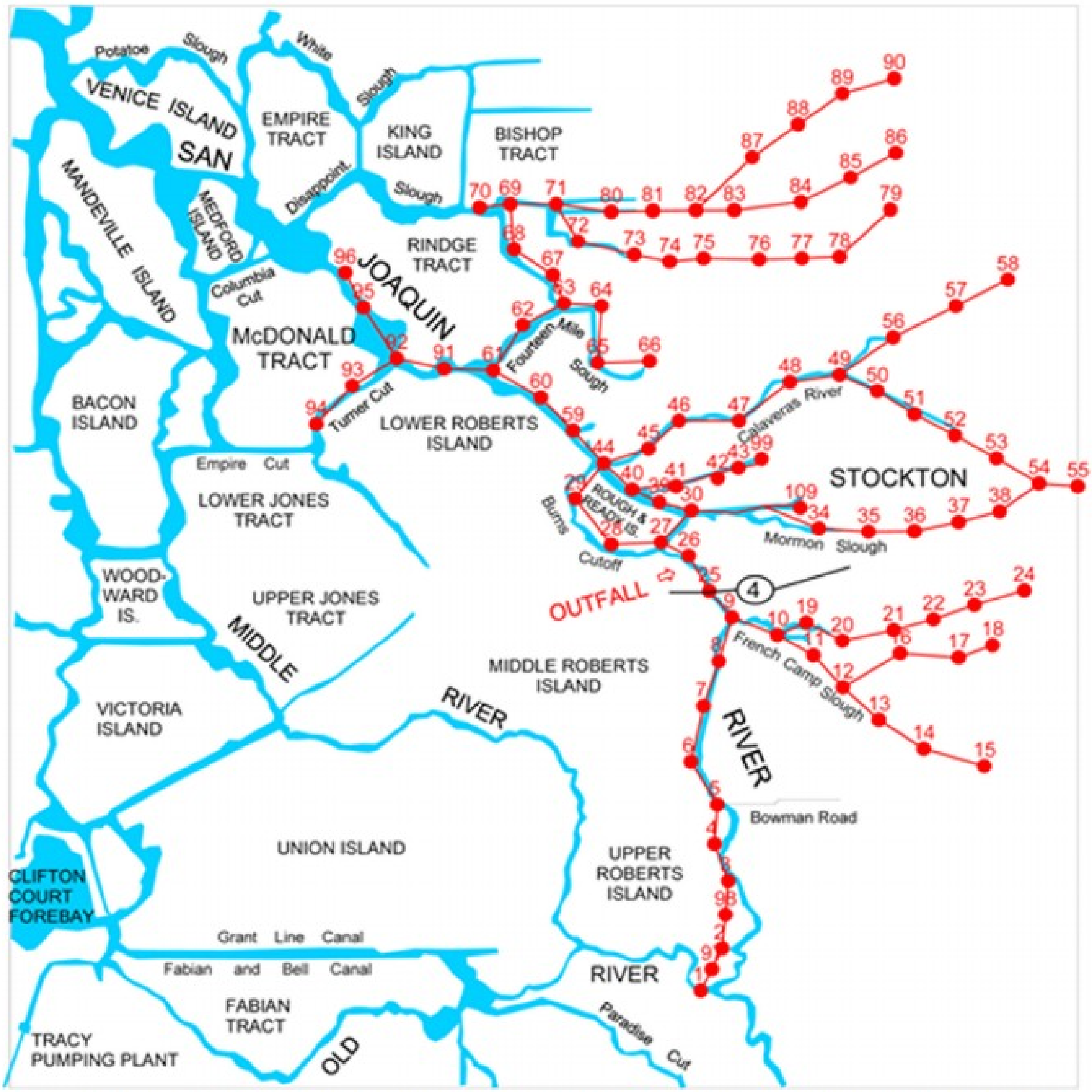

Fig. 2 San Joaquin River Link-Node model domain. Rough and Ready Island (RRI) monitoring station is at *node 40*. Garwood Bridge is between *nodes 25* and *26*

The model was originally developed for the City of Stockton (Schanz and Chen 1993) and was later used for the Interim South Delta Program (Chen and Tsai 1997), CALFED program (Chen and Tsai 2002), and SJR DO TMDL (Systech 2008). The model was adapted from Chen and Orlob (1975), which was based on Feigner and Harris (1970). The Chen and Orlob (1975) model led to development of WASP5. Chen and Tsai (2002) refined the model to include detritus and its sedimentation, resuspension, and oxygen demand. To more accurately represent photosynthesis in the DWSC, phytoplankton growth is simulated separately and limited to the top of the water column where sunlight penetrates (Chen and Tsai 2002).

Boundary conditions

Boundary conditions consisted primarily of observed data since previous modelers reported problems using model output to define boundary conditions (Monismith et al. 2008; HydroQual 2006a). Input data consisted of tidal stage, channel bathymetry, meteorological data, point source water quality and flow data, and water quality and flow data for the upstream SJR. Tidal stage, flow, and continuously monitored water quality data sets were obtained from the USGS and California DWR (CDWR 2013a). Water surface elevation data were obtained for the following stations (Fig. 1): Turner Cut (USGS No. 11311300), Rough and Ready Island (DWR Station RRI), and Venice Island (DWR Station VNI). High-frequency water quality monitoring data for temperature, DO, chlorophyll *a*, and turbidity were obtained for the RRI station. Meteorological data were obtained from a Stockton weather station and consisted of dry bulb temperature (°C), dew point temperature (°C), cloud cover, air pressure (millibar), wind speed (m s^{-1}), wind speed (miles h^{-1}), wind direction (degrees from north), evaporation rate (in. $month^{-1}$), and precipitation rate (in. $month^{-1}$) (CDWR 2013b). River channel dimensions were calculated using a combination of satellite imagery available in Google Earth and a GIS-based digital elevation model created by the USGS (CDWR 2007). Discharge compliance reports were used to obtain flow and water quality data for the RWCF effluent (CVRWQCB 2008). Flow and water quality for the sloughs, creeks, and rivers were created using a combination of observed data—including data collected as part of the DO TMDL project (Stringfellow and Camarillo 2014)—and simulation results (where observed data were not available) from the Sacramento River application of the watershed model WARMF (Herr et al. 2010). Measured water quality and flow data were used exclusively to define the upstream SJR input. The upstream model boundary was placed downstream of the confluence with Old River to allow for proper accounting of river diversions based on observed flow. Water quality data used included that for carbonaceous biochemical oxygen demand (CBOD, mg L^{-1}), DO (mg L^{-1}), ammonia-N (mg L^{-1}), nitrate-N (mg L^{-1}), phosphate-P (mg L^{-1}), chlorophyll *a* (μg L^{-1}), specific conductance (SpC, μmho cm^{-1}), total suspended solids (TSS, mg L^{-1}), and volatile suspended solids (VSS, mg L^{-1}). Sample collection and analytical descriptions are contained in previous publications (Stringfellow and Camarillo 2014; Stringfellow 2008b). Standard methods for analysis were used to the extent possible.

Model calibration

The SJR-Link-Node model was calibrated for the calendar years 2005 to 2010, a period that includes high flow and low flow conditions in the SJR (CDWR 2013a). Calibration was first completed for water surface elevation and flow and then for temperature and water quality (Sheeder and Herr 2013). Flow data from the USGS Garwood Bridge station (USGS No. 11304810) and high-frequency DO data from the DWR RRI station were used for calibration. Since the flow is tidal, net hourly flow was calculated using a 25-h moving average ($Q_{DWSC,i}$, m^3 s^{-1}), centered on the ith hour (Eq. 1). Flow data were 89 %

complete; missing data were largely the result of equipment malfunction during flooding in 2005–2006.

$$Q_{DWSC,i} = \frac{1}{25}\sum_{j=i-12}^{i+12} Q_{DWSC,j} \tag{1}$$

Calibration was performed to maximize the coefficient of determination (R^2) and to minimize mean and absolute errors (Eqs. 2 and 3). Average hourly DO data were 95 % complete.

$$\text{Mean Error} = \sum_{i=1}^{n} \frac{DO_{pred,i} - DO_{meas,i}}{n} \tag{2}$$

$$\text{Absolute Error} = \sum_{i=1}^{n} \frac{|DO_{pred,i} - DO_{meas,i}|}{n} \tag{3}$$

Model errors were calculated using $DO_{pred,i}$ as the ith modeled DO concentration at RRI (mg L^{-1}) and $DO_{meas,i}$ as the ith observed DO concentration at RRI (mg L^{-1}), such that there are n hourly data pairs. Relative mean and absolute errors were calculated by dividing values obtained from Eqs. 2 and 3 by the mean observed DO.

To improve model fit, simulations were run to evaluate model sensitivity to parameters affecting DO: temperature, ammonia, CBOD, and phytoplankton (Sheeder and Herr 2013). The analysis revealed that the aeration adjustment factor and soluble CBOD decay rate (Table S1) had the greatest impact on DO, and therefore, these coefficients were used for model calibration. In a previous model application, Chen and Tsai (2002) found that the model was most sensitive to (1) temperature correction factors for nitrification and BOD decay and (2) river mass loads. Chen and Tsai (2002) concluded that model sensitivity to temperature was not a concern since the model predicted temperature well. However, the sensitivity of the model to river mass loads—also noted by HydroQual (2006b)—led to studies to better define upstream SJR contributions and the work here uses those subsequently collected data sets (Stringfellow 2008b).

Model simulations

Four scenarios were run using the calibrated baseline model: (1) modification of the channel depth from its excavated depth of 10.7 m to its pre-existing natural depth of 3.8 m ("DWSC" scenario); (2) elimination of ODS mass loads from the upstream SJR ("SJR" scenario); (3) elimination of ODS mass loads from the Stockton RWCF ("RWCF" scenario); and (4) elimination of ODS mass loads from the urban tributaries discharging into the DWSC ("Tribs" scenario). The ODS consisted of BOD, ammonia-nitrogen, coliform bacteria, and chlorophyll *a*. In scenarios 2–4, the flows were held constant while only the ODS mass loads were removed. In the Tribs scenario, mass loads of ODS were removed from Fourteen Mile Slough, Bear Creek, Mosher Slough, Calaveras River, Mormon Slough, Duck Creek, Littlejohns Creek, French Camp Slough, Pixley Slough, and City of Stockton urban runoff (Fig. 2).

Excess net oxygen demand

Average hourly ENOD (in kg day^{-1}) was calculated when DO was below the standard (Eq. 4).

$$\begin{aligned}\text{average hour ENOD}_i = {} & (DO_{obj} - DO_{\text{meas},i}) \\ & \times (|Q_{\text{DWSC},i}| + 1.13) \\ & \times \left(\text{kg}/10^3\text{g}\right) \\ & \times \left(86,400\ \text{s}/\text{day}\right) \qquad (4)\end{aligned}$$

The regulatory standard (DO_{obj}) is 6 mg L^{-1} from September to November and 5 mg L^{-1} during all other months, $Q_{DWSC,i}$ (m^3 s^{-1}) is defined in Eq. 1, and 1.13 m^3 s^{-1} is the margin of safety used to account for flow measurement error (Gowdy and Grober 2005). The calculation of ENOD in Eq. 4 differs from that presented in Gowdy and Grober (2005) in that the absolute value of the net flow is used to include negative flow data. Net flow was negative—indicating a predominant upstream flow direction—for 7.5 % of observed hourly flow data. To calculate daily ENOD, the hourly ENOD values (in kg day^{-1}) were summed over 24 h (Eq. 5).

$$\begin{aligned}\text{average day ENOD} = {} & \frac{1}{24}\sum_{i=1}^{24} (DO_{obj} - DO_{\text{meas},i}) \\ & \times (|Q_{\text{DWSC},i}| + 1.13) \\ & \times \left(\text{kg}/10^3\text{g}\right) \\ & \times \left(86,400\ \text{s}/\text{day}\right) \qquad (5)\end{aligned}$$

Hourly ENOD is only included in the summation when DO is below the standard. In evaluating output from the model scenarios, $DO_{meas,i}$ was replaced with $DO_{pred,i}$ in Eqs. 4 and 5.

Results and discussion

Model calibration and uncertainty analysis

Model calculations for water elevation, flow, and temperature were first verified (Sheeder and Herr 2013). The comparison of simulated and observed water elevations at RRI yielded $R^2 = 0.90$, indicating that the model is adequately simulating hydrodynamics. With the exception of diurnal peaks, most flow data were adequately represented. Inaccurate peak prediction was likely due to inaccuracies in the bathymetry data although the USGS DEM used here is the best available (Sheeder and Herr 2013). Mean and absolute errors for temperature at RRI were −0.7 and 0.9 °C, respectively, indicating that the model adequately simulates temperature.

Model uncertainty for DO was tested by varying parameters to observe the impact on simulated DO (Sheeder and Herr 2013). Parameters tested were net flow at RRI, nitrification rate, soluble CBOD decay rate, and phytoplankton death rate. An increase in net flow of 14 $m^3 s^{-1}$ resulted in a modest mean increase of DO of approximately 1.3 %. Increasing the nitrification rate from 0.05 to 0.15 day^{-1} resulted in a 3 % decrease in mean DO. Increasing the soluble CBOD decay rate from 0.15 to 0.30 day^{-1} resulted in a mean DO increase of 0.1 mg L^{-1}. The model results were not sensitive to the phytoplankton death rate, as indicated by a decrease in DO of less than 1 % occurring as the result of a 300 % increase in the phytoplankton death rate. Based on this analysis, the soluble CBOD decay constant was used for model calibration along with the aeration adjustment constant that directly controls how much oxygen enters the SJR from the atmosphere (Sheeder and Herr 2013).

Model calibration was done to provide the best fit for observed hourly DO data (*n* = 50,434) based on minimizing model errors (Fig. 3). A soluble CBOD decay coefficient of 0.30 day^{-1} and an aeration adjustment factor of 1.8 were selected along with other coefficients (Table S1). Mean and absolute errors were −0.11 and 0.90 mg L^{-1} (1.4 and 12 %, relative mean and relative absolute errors), respectively, indicating that the model provided a good fit for hourly DO. As a comparison, in a 1-D model study of the Neuse River Estuary (NC), Bowen and Hieronymus (2003) determined a mean error of −0.324 mg L^{-1} for DO based on calibration with 3 years of data at three stations (*n* = 16,586). In a 3-D study of the Neuse River, Wool et al. (2003) reported relative mean errors of −1.21, 0.11, 5.73, and 12.08 % at four stations (*N* = 236–273). In a 21-year simulation of the Chesapeake Bay, the relative mean error for DO was 13.5 % (Cerco and Noel 2013). In a 1-D study of the South Umpqua River (OR), Turner (2009) reported mean DO error of 0.5 mg L^{-1} and absolute mean error of 1.0 mg L^{-1}. Using a 3-D model of the DWSC for years 2000–2001, HydroQual (2006a) reported DO mean errors of 0.22 and 0.14 mg L^{-1} (3.30 and 2.18 % relative mean errors) at two stations. Based on these comparisons, the SJR-Link-Node appears to be adequately simulating DO.

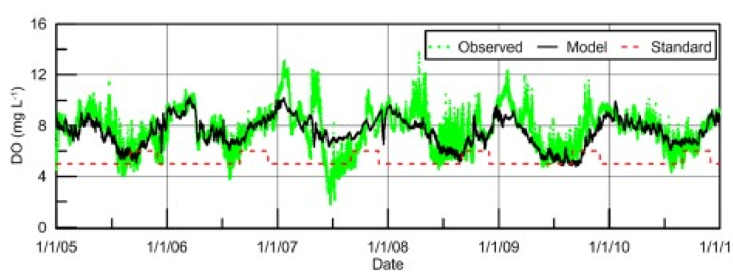

Fig. 3 San Joaquin River dissolved oxygen (DO) data paired with hourly SJR-Link-Node output. The standard is 6 mg L^{-1} from September to November and 5 mg L^{-1} during other months

Although average DO values were predicted well, the model was less efficient in predicting peaks and troughs (Fig. 3). The DO was over-predicted when DO was low and under-predicted when DO was high. The annual variability in model errors was consistent (with mean annual model residuals of ±1 mg L^{-1}), although model errors for the year 2007 demonstrated the most variability (Fig. 4). A possible explanation was that there was less precipitation and flow in 2007 than in the two previous years (Fig. 5), which may have affected storage of ODS in the river basin. In 2007, DO was over-predicted, while it was under-predicted in 2009 (Fig. 4). Violations of the regulatory standard were observed during 4017 h over the 6-year period (8 % of all hours), resulting in 286 days with violations (13 % of all days). The model predicted violations during 2007 h (4 % of all hours), distributed over 102 days (5 % of all days). Violations were predicted from July through October, while observed violations occurred during all months except February, March, and April (Table 1). These results suggest that the model is more efficiently predicting violations in the summer than in the winter, which is preferred since DO impairment primarily occurs during the summer.

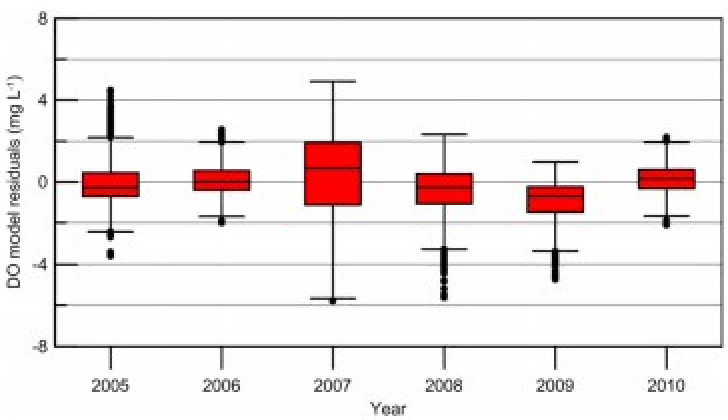

Fig. 4 Dissolved oxygen (DO) model residuals by year

Fig. 5 San Joaquin River hourly flow data at Garwood Bridge

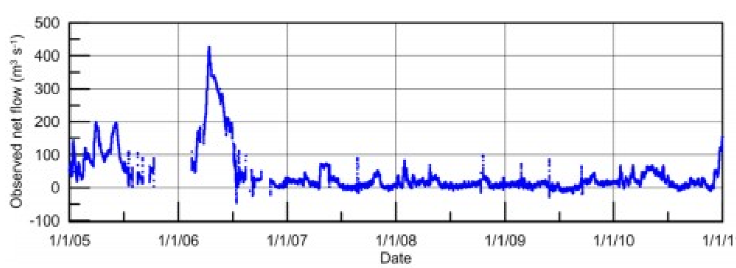

Table 1 Hourly violations observed at Rough & Ready Island (RRI) and predicted using the San Joaquin River Link-Node model (2005–2010)

Month	Observed at RRI	Baseline model[a]	Model simulation scenarios[b]			
			"DWSC"	"SJR"	"RWCF"	"Tribs"
Jan	5	–	–	–	–	–
Feb	–	–	–	–	–	–
Mar	–	–	–	–	–	–
Apr	–	–	–	–	–	–
May	29	–	–	–	–	–
Jun	521	–	–	–	–	–
Jul	891	87	–	40	34	9
Aug	548	66	–	8	20	5
Sep	1536	1579	758	920	1041	1540
Oct	408	275	–	–	189	212
Nov	77	–	–	–	–	–
Dec	2	–	2	–	–	–
Sum	4017	2007	760	968	1284	1766

[a] Model calibrated using data from 2005 to 2010

[b] Simulations based on (1) natural channel depth instead of dredged depth ("DWSC"), (2) elimination of upstream mass loads ("SJR"), (3) elimination of the Regional Wastewater Control Facility mass loads ("RWCF"), and (4) elimination of mass loads from urban tributaries ("Tribs")

Model simulations

All four model scenarios resulted in DO improvement compared with the baseline model (Fig. 6). Increases in DO ranged from 0 to 2.0 mg L^{-1}. In some cases, the DWSC scenario resulted in lower DO than what was predicted using the baseline model, as indicated by negative values in Fig. 6, although these cases were infrequently observed. The total number of DO violations predicted was also reduced in the four scenarios (Table 1). The largest improvement in DO was observed for the DWSC scenario, where 62 % fewer hourly DO violations were predicted. The SJR scenario had the second largest improvement in DO with 52 % fewer hourly DO violations predicted. Improvement observed for the RWCF and Tribs scenarios were smaller with 36 and 12 % fewer hourly DO violations, respectively. Regardless of whether DO concentration, violations, or ENOD is used as a metric, the DWSC scenario still yielded the most improvement in simulated DO followed by the improvements observed in the SJR, RCWF, and Tribs scenarios (Table 2). Despite the improvement observed for all four scenarios, eliminating any one factor did not result in complete resolution of low DO.

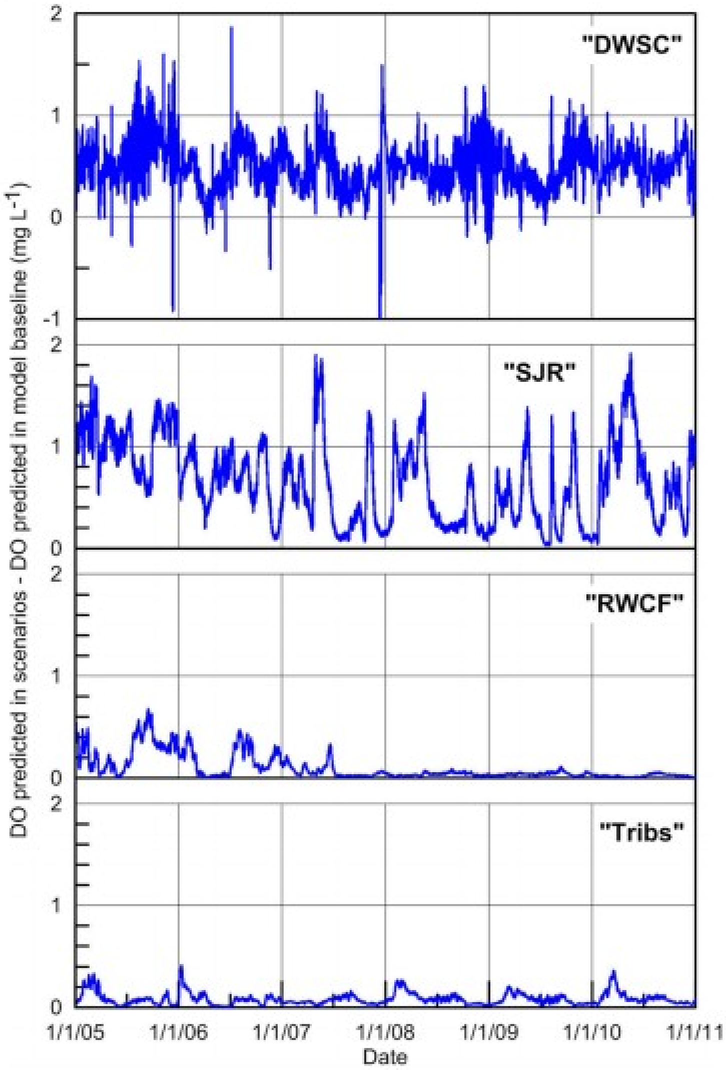

Fig. 6 Predicted increases in dissolved oxygen (DO) relative to the baseline model (hourly predictions)

Table 2 San Joaquin River Link-Node model predictions for dissolved oxygen (DO) and excess net oxygen demand (ENOD), 2005–2010

Parameters	Baseline model[a]	Model scenarios[b]			
		"DWSC"	"SJR"	"RWCF"	"Tribs"
DO ($mg\ L^{-1}$)[a] when violations[c] predicted by baseline model, $n = 2007$[d]	5.43 ± 0.34	6.08 ± 0.53	5.85 ± 0.54	5.68 ± 0.49	5.53 ± 0.33
Number of hours with ENOD[c,e]	1409	628	811	1091	1168
95th percentile for hourly ENOD (kg day^{-1})[e]	1922	1256	1686	1822	1775
Max hourly ENOD (kg day^{-1})[e]	4925	2648	4255	4389	4349
Number of days with ENOD[c,e]	79	31	46	60	63
95th percentile for daily ENOD (kg day^{-1})[e]	1866	1176	1723	1790	1746
Max daily ENOD (kg day^{-1})[e]	2899	1344	2492	2577	2538
Reduction in total summed ENOD relative to baseline	–	69 %	49 %	20 %	19 %

[a] Model calibrated using data from 2005 to 2010

[b] Simulations based on (1) natural channel depth instead of dredged depth ("DWSC"), (2) elimination of upstream mass loads ("SJR"), (3) elimination of the Regional Wastewater Control Facility mass loads ("RWCF"), and (4) elimination of mass loads from urban tributaries ("Tribs")

[c] DO standard is 5 mg L^{-1} except September through November when it is 6 mg L^{-1}

[d] Mean ± standard deviation

[e] ENOD is calculated when predicted DO is below standard and flow data are available

During the study period, mass load contributions of ammonia-nitrogen and CBOD5 from the RWCF decreased as the result of facility upgrades (Fig. 7), so that the ODS contribution from the RWCF discharge also went down (Fig. 6). When only the model results following the RWCF upgrade are considered (completed by August 1, 2007), the distribution of DO violations caused by ODS mass loads is as follows: 55 % from the SJR, 27 % from the urban tributaries, and 18 % from the RWCF. If all predicted hourly DO violations caused by the four factors are considered together, the relative sources of DO impairment caused by the DWSC channel depth, SJR ODS mass loads, RWCF ODS mass loads, and urban tributary ODS mass loads are as follows: 38, 32, 22, and 8 %, respectively. Excluding the period before the RWCF upgrade, these values are 45, 30, 10, and 15 %. The results indicating shared responsibility are not definitive and are a function of the model and data used (e.g., selection of the study period). Additional verification is warranted.

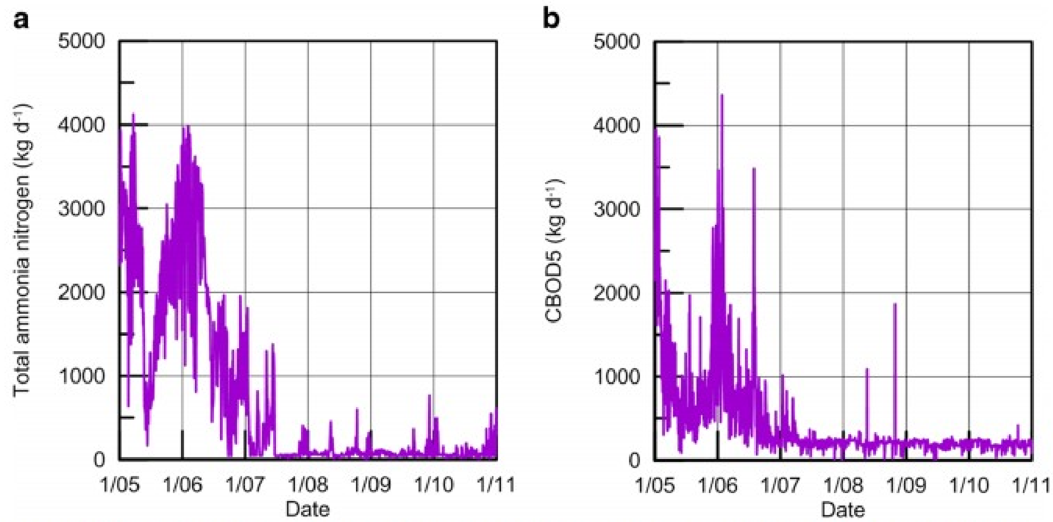

Fig. 7 Daily mass loads of **a** total ammonia nitrogen (ammonium and aqueous ammonia) and **b** 5-day carbonaceous biochemical oxygen demand (CBOD5) from the Regional Wastewater Control Facility (RWCF)

Ship channel impact

The DWSC scenario resulted in DO increases that varied from 0 to 1.0 mg L^{-1} (Fig. 6). The largest improvements were predicted during early winter (November and December), and the smallest improvement were predicted during spring (April) (Table S2). However, DO increases in the DWSC scenario were sufficiently high during the summer and fall to eliminate predicted violations in July, August, and October (Table 1). Approximately half of the predicted DO violations in September—when the DO standard is increased to 6 mg L^{-1}—were eliminated as the result of this scenario. Year-to-year variability in predicted improvements was apparent. Improvements were consistent throughout most of 2010, higher in late summer/fall of 2005, and higher in the winter of 2008 and 2009 (Fig. 6).

Channel depth has several effects on DO. Improvements predicted for the DWSC scenario demonstrate the important relationship between retention time (decreased due to decreased channel depth) and exerted oxygen demand. The DWSC scenario was more beneficial in November and December when river flows were low (due to fewer reservoir releases) and less beneficial in April when river flows were high (Figs. 5 and 6; Table S3). Low flows and the resulting long retention times are causative factors for eutrophication in rivers and estuaries (Hilton et al. 2006). However, high flows can result in transport of high mass loads of nutrients into estuaries that can also simulation phytoplankton growth and decay (Cerco and Noel 2013). In the SJR, low river DO occurs when the river flow is low, but the relationship between DO and river flow is difficult to discern (Gowdy and Grober 2005). During the study period, when minimum daily DO was less than 5 to 6 mg L^{-1}, net flow was typically less than 100 $m^3\ s^{-1}$; however, low flows were not synonymous with low DO (Fig. 8). Explanations for this unexpected lack of correlation between flow and low DO include flow manipulation within the river (e.g., reservoir releases and river diversions) and productivity that is light-limited rather than nutrient-limited. Complex relationships between flow and DO have been noted in other systems, likely the result of confounding variables (Cerco and Noel 2013). Reducing channel depth also increased atmospheric oxygen inputs due to the increased surface area to volume ratio. Decreased channel depth increased the portion of the water column that is illuminated, increasing phytoplankton activity and the resulting photosynthesis and oxygen production. In the SJR, phytoplankton growth is thought to be seasonably limited by light and not nutrients (Jassby 2005; Leland 2003), which is common in eutrophic rivers (Hilton et al. 2006). Variability in predicted DO improvement for the DWSC scenario is likely caused by the interdependence of these multiple factors although the overall effect is a net increase in DO.

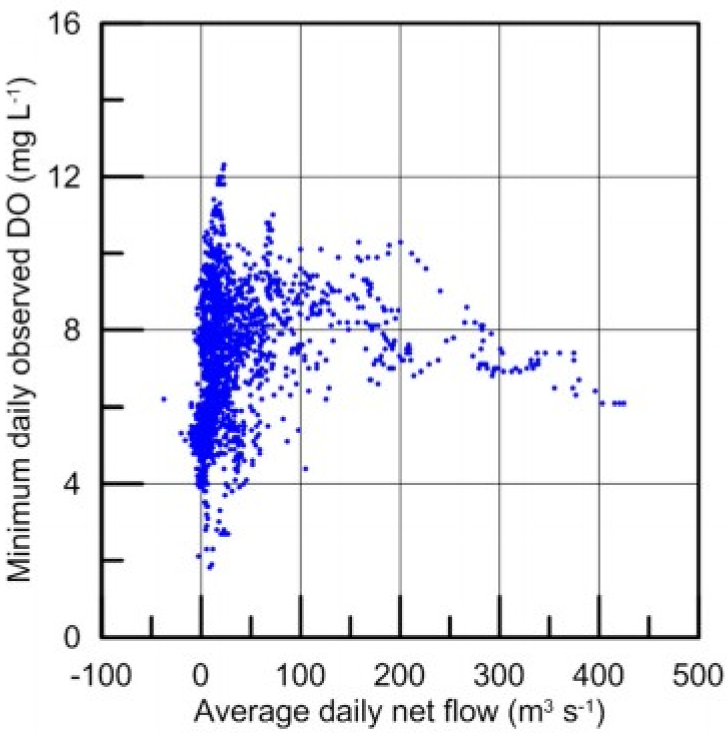

Fig. 8 Relationship between river flow and minimum dissolved oxygen (DO)

Watershed impact

The SJR scenario resulted in increased DO of 0 to 2.0 mg L^{-1} although the improvement was variable (Fig. 6). The largest DO improvements occurred in May, while the smallest DO improvements were in August and September (Table S2). A reduction in predicted DO violations occurred from July through September, and the prediction of DO violations in October was eliminated (Table 1).

The ODS mass loads from the SJR were influenced by season and flow rate among other factors. In the SJR scenario, the impact of removing ODS mass loads from the SJR was most influential in May when there is agricultural activity in the watershed (during the growing season) and flow rates are sufficient to transport large quantities of ODS to the DWSC (Table S3). May is not a time of year when phytoplankton growth is most significant (Volkmar and Dahlgren 2006; Volkmar et al. 2011), suggesting that both flow and phytoplankton growth affect low DO in the DWSC. In the summer when the phytoplankton production is high (possibly due to increased water temperature), flows are low and transport of ODS is not significant, so the effect of the SJR scenario appears lower (Fig. 6). The co-existing effects of ODS and flow on downstream DO suggest that storage of watershed-derived contaminants and subsequent release should be considered in determining the watershed impact. Establishing linkages between causative factors and the resulting eutrophication is difficult (Howarth et al. 2002), making modeling approaches such as the one taken here advantageous.

DO improvements in the SJR scenario were variable and timed differently than in the DWSC scenario (Fig. 6). The SJR scenario was most beneficial in 2005 and 2006 when the river was flooding and transport of ODS mass loads was higher than normal, likely due to erosion (Fig. 6). In 2007, 2008, and 2010, the SJR scenario resulted in improvement during the early part of the year. Although improvements were typically lower in the summer, a large improvement was predicted during the summer of 2007, a dry year. Reduction of watershed-derived ODS was more influential during this dry year, possibly the result of increased irrigation. The results here indicate that factors other than flow are important in DO outcomes.

The types of ODS present are likely important. In the summer, much of the ODS is derived from phytoplankton (Volkmar and Dahlgren 2006). Ammonia is also thought to significantly contribute to ODS in the SJR Estuary (Lehman et al. 2004; Jassby and Van Nieuwenhuyse 2005). In their modeling study, HydroQual (2006b) found that non-algal carbon sources (e.g., terrestrial carbon) are significant sources of DO impairment, resulting in 2.0–2.5 mg L^{-1} of DO consumption. Additionally, water temperature can have a compounding effect on the chemical and biological processes resulting from the impacts of ODS (Cloern 2001). The net effect is that the impact of the SJR mass loads on the estuary is highly variable (Fig. 6).

Wastewater treatment plant impact

The RWCF scenario resulted in increases in DO that ranged from 0 to 0.7 mg L^{-1} (Fig. 6). Improvements in DO primarily occurred in the first half of the study. Following a facility upgrade in 2007—which included a nitrification facility and free surface constructed wetlands—model results indicate that the discharge had little impact on DO (CVRWQCB 2008). The reduced impact of the RWCF was likely due to decreased ammonia and CBOD mass loads from the facility following the 2007 upgrade (Fig. 7). Prior to the upgrade, average daily ammonia-nitrogen mass loads were as high as 4000 kg day^{-1}. Following the upgrade, average daily ammonia-nitrogen mass loads were typically less than 500 kg day^{-1} (Fig. 7a). After the upgrade, effluent ammonia-nitrogen remained high in the winter (e.g., above 100 kg day^{-1}), corresponding with a period of low temperatures that likely affect the biological nitrification system (biotowers). However, DWSC DO in the winter months appears less sensitive to RWCF ODS mass loads, as evident by the lack of improvement when the loads are removed (Fig. 6).

In addition to ammonia, CBOD5 mass loadings from the RWCF also decreased after the facility upgrade. Prior to the upgrade, the effluent average daily CBOD5 mass loads were as high as 4500 kg day^{-1}, and after the upgrade, the CBOD5 mass loads were lower than 100 kg day^{-1}, with effluent CBOD5 typically at or near detection limits (Fig. 7b). Two occurrences of high CBOD5 and high ammonia nitrogen mass loads were evident in 2008. The causes of these events are unknown, but likely related to process upsets or mechanical failures.

Wastewater effluents often contribute to river water quality impairments and are part of TMDL waste load allocations (Pickett 1997; Turner et al. 2009). Here, it was possible to determine the water quality impact of a major facility upgrade, which demonstrates the functionality of an iterative TMDL feedback process and the value of consistent and continuous data collection (Chapra 2003). Although the facility upgrade improved water quality in the SJR, the improvement was subtle and difficult to discern from the highly variable DO concentrations. Following the upgrade, DO observed at RRI was 7.59 ± 1.54 mg L^{-1} (n = 29,403), which is not significantly different than the 7.67 ± 1.85 mg L^{-1} (n = 21,031) measured before the upgrade. However, the number of violations decreased following the upgrade from 10.5 % of hourly observations to 6.2 %, indicating that DO increased when water quality conditions were most critical.

The minor impact of ammonia-nitrogen from the Stockton RWCF was not expected based on the results of previous monitoring studies (Lehman et al. 2004; Jassby and Van Nieuwenhuyse 2005), but was predicted in a modeling study (HydroQual 2006a). HydroQual (2006b) predicted that elimination of ammonia-nitrogen from the Stockton RWCF would result in less than 1 mg L^{-1} increase in DWSC DO. The results in Fig. 6 are in agreement with that prediction.

Urban stormwater impact

The Tribs scenario resulted in DO improvements that ranged from 0 to 0.4 mg L^{-1} (Fig. 6). Increased DO was mostly predicted in the early months of the year (March) when storms occur. Less improvement was predicted where the observed flow was low (October and December). These patterns are supported by previous observations. Lee and Jones-Lee (2003) noted that urban stormwater was a significant source of ODS during a large storm in November 2002. However, even during 2005 and 2006 when there was flooding in the SJR, the improvement predicted in DO for the Tribs scenario was less than 0.4 mg L^{-1}. It is possible that a “first flush” phenomenon is occurring where the storms occurring following a long period of time without rainfall release large quantities of pollutants, while storms in the middle of a rainy season have lower pollutant concentrations. Data collected by our research group in 2011 and 2012 (unpublished) indicates that 10-day BOD in samples collected year-round from the urban tributaries can be high (up to 25 mg L^{-1}), but values are typically less than 10 mg L^{-1}. In samples collected at eight tributary locations (15–20 samples per site), the mean 10-day BOD values ranged from 0.92 to 9.88 mg L^{-1}. Flow data for these small tributaries is lacking although flow is likely minimal outside of storm events, since they drain small catchments and the annual average rainfall for the area is low (approximately 35 cm). Based on model results, the effect of ODS mass loads from urban tributaries appears low relative to other factors considered here. The effects of urban surface water runoff are often difficult to delineate from other forms of non-point source pollution (Howarth et al. 2002). Further water quality and flow data collection could confirm the impacts of ODS from urban tributaries.

Model limitations

Similar to other DO models, SJR-Link-Node is formulated using a combination of observed data, bathymetric representations, biological/chemical processes, and model parameters (Vellidis et al. 2006). The availability and quality of existing data represent significant limitations here and in other models (Chapra 2003). Observational data has inherent variability as well as spatial and temporal limitations (especially grab sample data). Here, continuous DO measurements represent a rich time series data set although the data were collected at a single location (RRI). Error was introduced into the model by errors in flow measurement, water quality tests, and bathymetric data.

The 1-D model represents another limitation in that water quality parameters are averaged over river segments and river hydrodynamics are simplified. The simple 1-D model was used because it is better suited for the available data, which is important in model selection (Chapra 2003). One-dimensional models are used extensively in TMDL management (Vellidis et al. 2006) and are used extensively throughout the Delta to investigate management scenarios (e.g., CDWR 2013c). The 1-D model allowed for modeling over a long time period with a small time step. The model was not intended for use in forecasting; models used for forecasting require more rigorous verification. The results here suggest that the modeling approach taken is sound, producing error metrics comparable with those from other projects (HydroQual 2006b; Bowen and Hieronymus 2003; Wool et al. 2003; Turner et al. 2009; Cerco and Noel 2013).

ENOD as an evaluation metric

Calculated ENOD varies as a result of variability in DO and flow (Table 2, Figs. 3 and 5). Some of this variability is seasonal and related to water demand in the agricultural watershed, while diurnal variability in DO is related to diel phytoplankton cycles driven by light. Although ENOD was predicted to occur on 79 days in the baseline model, ENOD was not consistently predicted during all hours of the day (Fig. 9). ENOD was predicted to occur during all 24 h of the day for 43 of the 79 days (54 %). As shown in Fig. 9, ENOD was more likely predicted in the early morning hours (midnight to 8 a.m.) and less likely predicted in the afternoon and evening (noon to 9 p.m.). A reasonable explanation for ENOD to occur during the early morning hours is that phytoplankton respiration is occurring while photosynthesis is not.

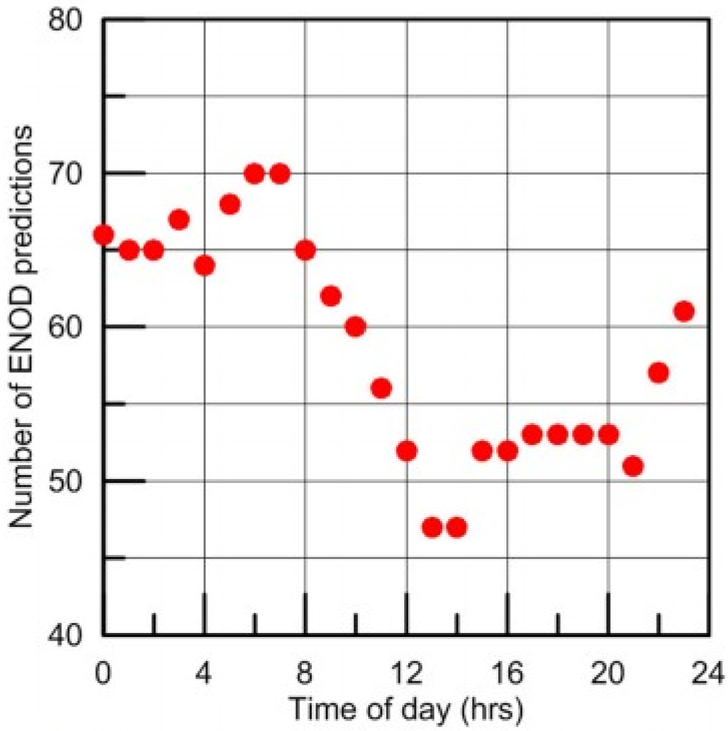

Fig. 9 Predicted excess net oxygen demand (ENOD) over the hours of the day

The relationship between predicted ENOD and DO deficit was not consistent (Fig. 10), suggesting that DO and flow independently affect ENOD. One advantage of using ENOD over DO is that it is only calculated when DO is below the standard. Additionally, ENOD is a measure of DO deficit on a mass rate basis, providing an estimate of volumetric DO impairment and the supplemental aeration needed to meet the standard. An advantage of using ENOD is that it could provide the basis of design for supplemental aeration systems and estimates of operating times. Calculations of ENOD could also be used to determine responsibility for operating the aeration system.

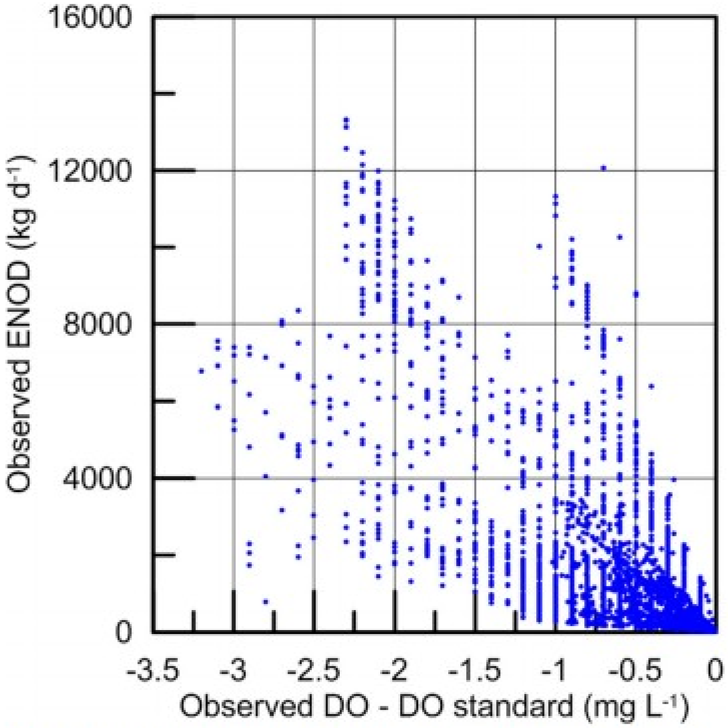

Fig. 10 Relationship between excess net oxygen demand (ENOD) and dissolved oxygen (DO) deficit

A drawback of ENOD is that if the flow rate is zero, the ENOD will also be zero, even if the DO deficit is high. Low flow rates will cause ENOD to be low. Since both flow rate and DO are measured at a single point, the absence of flow does not mean that a large volume of water is not below the DO standard. Monitoring DO and flow at a single point is only effective for developing volumetric estimates of DO deficit if the net flow is sufficiently high to identify an entire "slug" of low DO water as it travels past the monitoring point. If net flow is at or near zero, it is not possible to determine the volume of impaired river water using a single monitoring point because the impaired water is stagnant within the DWSC. In lieu of ENOD, it may be more relevant to determine volumetric estimates of DO deficit, representing the volume of water impacted by low DO since the supplemental aeration system is stationary and its radius of influence is limited.

The deployment of additional DO monitoring stations would allow for 3-D estimates of low DO zones within the DWSC, which may be more useful. Volumetric estimates of low DO water are calculated in the Chesapeake Bay using a 3-D estuary model that allows for evaluation of anoxic volumes in response to watershed-derived mass loads (Bever et al. 2013; Cerco and Noel 2013). Given that flow in the DWSC can be low and that low DO concentrations in the DWSC tend to be widely spread and not isolated (Schmieder et al. 2008; Spier et al. 2013), a volumetric approach is warranted.

Implications for the TMDL

Identifying the sources of impairment is an essential part of the TMDL process and is needed for waste load allocation (US EPA 2008; Vellidis et al. 2006). The extensive monitoring and modeling that has been done in the SJR basin is similar to what is done in other watershed-wide TMDL efforts (US EPA 2008; Shenk and Linker 2013). Previously, the SJR-Link-Node estuary model was connected with a watershed model to identify sources of watershed-derived ODS (Gulati et al. 2016; Stringfellow and Camarillo 2014). Monitoring efforts continue. Monitoring and modeling serve as the bases for limiting point-source discharges and mitigating diffuse sources (e.g., agriculture) by best management practices (Bowen and Hieronymus 2003; Wool et al. 2003; Turner et al. 2009). Here, the channel depth is a source of impairment, suggesting that the channel should be restored to its original depth since it is not possible to implement source controls. However, the economic importance of the DWSC to the regional economy suggests that channel restoration is unlikely. Operation of a supplemental aeration system is a good alternative (Speece 1996; Melching et al. 2013), and such a system is currently in place (Jones & Stokes 2005b). The results of the SJR-Link-Node can be used to assign responsibility for that system.

The work here can also be used to guide future modeling studies, which would benefit from additional data sets. Since 2008, DO measurements have been taken at three depths (1, 3, and 6 m depth) at the RRI, and these data could be used in future studies. If 2-D or 3-D models are desired, more extensive data—with better geospatial coverage—will be needed. The trade-offs between investment in data collection and better resolution in the result need to be investigated. If use of the SJR-Link-Node continues, efforts to improve the model fits for DO peaks and troughs are recommended. In future studies, additional scenarios could be investigated to consider how the various DO stressors interact. For example, how will reductions in upstream mass inputs reduce the impact of the ship channel?

Conclusion

Collection of continuous data and 1-D modeling made it possible to simulate DO in the complex and dynamic SJR Estuary. Continuous DO data were used to model diel growth and decay that dictate DO outcomes in the summer. The SJR-Link-Node model was successfully calibrated for a 6-year period that included both wet and dry years and where violation of the DO standard was frequent. Model errors for DO were comparable with values reported in previous studies. Simulation results suggest that the channel depth is most influential in low DO, followed by ODS from the agricultural watershed. Contributions from the RWCF and urban tributaries had smaller impacts on DO. Upgrading the RWCF significantly reduced its impact on DO. Verification of these results is recommended as is additional data collection and incorporation of multi-depth DO data.

Calculation of ENOD, the mass loading rate of oxygen deficit, was beneficial for estimating the quantity of low DO water transported through the DWSC. Deployment of additional continuous monitoring devices, further use of modeling, and additional surveys would further support development of volumetric-based assessment.

The results contribute to understanding the sources of DO impairment in the SJR DWSC and provide guidance to those investigating impaired water bodies that have altered channel geometry. The results, as presented, provide data that can guide TMDL decisions. This data could further be used for benchmarking and for evaluation of restoration projects, promoting a scientifically based feedback loop and, ultimately, improved water quality in the estuary.

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